CHAPTER 4

Source Water Protection: Its Role in Controlling Disinfection By-Products (DBPs) and Microbial Contaminants[1]

Introduction

Passage of the 1996 Safe Drinking Water Act Amendments (SDWAA) has focused the attention of water utility managers and public health and regulatory officials on source water protection (SWP) and its role in protecting public water supplies. There is growing awareness that water treatment and/or disinfection may not always be enough to ensure the provision of potable and safe water to the consumer. The 1993 cryptosporidiosis outbreak in Milwaukee, WI, has raised the possibility that even water suppliers which meet all of the Surface Water Treatment Rule (SWTR) requirements of the SDWA are vulnerable (Okun et al. 1997; Fox and Lytle 1996).

Most utilities in the U.S. invest a great deal of time, energy, and capital in developing mechanisms for protecting against the impact of sudden changes in influent water quality. Some of these mechanisms include investment in excess capacity and development of emergency procedures (Miller 1989).

Concern over source water protection is not limited to surface water supplies. Many ground water supplies have proven to be vulnerable as well, resulting in the various states implementing wellhead protection programs. Based on the 1996 amendments, the states will have to implement programs to decide if a system's source of supply is threatened as well as determine the means to prevent pollution. Communities will be allowed to ask for state assistance, and a certain percentage of the State Revolving Loan Fund has been earmarked to assist with source water protection (Howell 1997).

Water supplies vary greatly in the nature of the source water they use and in the circumstances under which they provide water to their customers. Nevertheless, there are some common elements that are applicable to source water protection in general. For example, land-use planning can provide information that is related to source water protection. Information on population densities, the ratio of pervious to impervious land cover, and the location of point and non-point sources of pollution can be important in assessing problems associated with both ground and surface source water protection.

As part of the Clean Water Act (CWA), Comprehensive River Basin Planning was initiated under Section 208 of the CWA. A major effort was undertaken to bring to bear the existing art and science of comprehensive planning in river basins with regard to minimizing the impact of point and non-point pollution on water quality in streams, lakes, and ground water. Many of the approaches suggested in studies developed under this program are very relevant to the issue of source water protection today.

Stream and contaminant transport models provide a mechanism for identifying and assessing the pollutants that are likely to be present in surface sources used for water supply. These models can be used for (1) identification of communities whose water supplies could be vulnerable to contamination resulting from industrial and municipal discharges or urban and agricultural runoff, (2) design of water and wastewater treatment plants, (3) design and implementation of water quality monitoring programs, and (4) other water resource planning efforts requiring information on the quality of surface waters (Clark et al. 1998).

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This chapter will explore SWP as outlined in the Safe Drinking Water Act (SDWA): the nature of threats to source water quality; methods, monitoring, and assessment of pathogens; technologies for control of water quality; the use of models to assess water utility vulnerability; and the relationship of source water protection to watershed management.

The Safe Drinking Water Act and Source Water Protection

The SDWA was passed in 1974 and amended in 1986 and 1996, but SWP under the SDWA actually began with the SDWA Amendments of 1986. The 1986 amendments included provisions for "Protection of Ground Water Sources of Water." Two programs were set up under this requirement: the "Sole Source Aquifer Demonstration Program," to establish demonstration programs to protect critical aquifer areas from degradation; and the "Wellhead Protection Program," which required states to develop programs for protecting areas around public water supply wells to prevent contamination from residential, industrial, and farming activities.

The SWTR, published in June 29, 1989, and effective December 31, 1990, was designed to prevent waterborne diseases caused by viruses, *Legionella*, and *Giardia lamblia*. These disease-causing organisms are present in varying concentrations in most surface waters. This rule requires water systems to filter and disinfect water from surface water sources to reduce the occurrence of unsafe levels of these microbes. Surface water is particularly susceptible to microbial contamination from sewage treatment plant discharges, storm water runoff, and snow melt. The rule sets nonenforceable health goals and maximum contaminant level goals (MCLGs) for viruses, *Legionella*, and *Giardia lamblia* at zero because any amount of exposure to these contaminants represents some health risk. In establishing legal limits for these contaminants in drinking water, the U.S. Environmental Protection Agency (EPA) can set either a maximum contaminant level (MCL), which is a legal limit, and require monitoring for the contaminant in drinking water, or, for those contaminants that are difficult to measure, EPA can establish a treatment technique requirement. Since measuring disease-causing microbes in drinking water is not considered to be feasible, EPA established a treatment technique in this rule.

The SWTR Guidance Manual (USEPA 1991) identifies both natural and human-caused sources of contamination to be controlled. These sources include wild animal populations, wastewater treatment plants, grazing animals, feedlots, and recreational activities. The Guidance Manual recommends that grazing and sewage discharges not be permitted within the watershed of unfiltered systems. Both may be permissible on a case-by-case basis where the watershed provides a long detention time and a high dilution between the location of the activity and the water intake. The nonfiltering utility is required to develop state-approved techniques to eliminate or reduce the effect of the identified point and non-point pathogenic contamination sources.

In the 1996 amendments to the SDWA, protection of source waters was given greater emphasis to strengthen protection against microbial contaminants, particularly *Cryptosporidium*, while reducing potential health risks due to disinfection by-products. This increased protection is embodied in the Interim Enhanced SWTR (IESWTR) (USEPA 1998). This rule applies to public water systems that use surface water or ground water under the direct influence of surface water (GWUDI) and serve at least 10,000 people. The final IESWTR (USEPA 1998), issued December 16, 1998, and effective February 16, 1999, includes several requirements specific to finished drinking water, and three that relate to watershed protection. EPA is to

- set a MCLG of zero for *Cryptosporidium*
- require a 2-log oocyst removal for drinking water systems that filter
- include *Cryptosporidium* in the watershed control requirements for unfiltered public water systems (Filtration Avoidance Criteria [FAC])

- require covers on new finished water reservoirs
- set other requirements that build upon the SWTR's treatment technique requirements

States are to

- conduct sanitary surveys for all surface water systems, regardless of size

The watershed control program for *Cryptosporidium* must identify watershed characteristics and activities that may have an adverse effect on source water quality and monitor the occurrence of activities that may have an adverse effect on source water quality. The state must determine whether the established watershed control program is adequate to limit potential contamination by *Cryptosporidium* oocysts.

The 1996 SDWA amendments also included four prevention approaches as part of establishing a new charter for protecting the nation's public water systems: SWP, State Ground Water Protection, Capacity Development, and Operator Certification. The SWP approach established a new Section 1453 for source water quality assessments. States with public water supply (PWS) primacy were required to submit source water assessment program plans for EPA approval. A state assessment program is required to (1) delineate the boundaries of the areas providing source waters for public water systems, (2) identify, to the extent practicable, the origins of regulated and certain unregulated contaminants in the delineated area, and (3) determine the susceptibility of public water systems to the identified contaminants. Assessments are to be completed for all public water systems within two years after EPA approval of the state's program. To avoid duplication, assessments may make use of sanitary surveys, state wellhead protection programs, pesticide state management plans, state watershed initiatives including efforts under the SWTR, and efforts under the Federal Water Pollution Control Act, commonly referred to as the CWA. Section 1453 provides a number of additional features that may be used to assist the state in promoting and developing SWP programs.

In support of the Microbial–Disinfection By-Products (M–DBP) rule-making process, the Information Collection Rule (ICR) was promulgated (May 14, 1996; 61 FR 24354; effective June 18, 1996) to collect occurrence and treatment information to help evaluate the need for possible changes to the current SWTR and existing microbial treatment practices, and to help evaluate the need for future regulation for disinfectants and disinfection by-products (D/DBPs) (USEPA 1996a). The ICR pertains to large public water systems serving at least 100,000 people, and a more limited set of ICR requirements pertain to ground water systems serving between 50,000 and 100,000 people. About 300 PWSs operating 500 treatment plants were involved in the extensive data collection required under the rule. Surface water systems were required to monitor for microbials, including bacteria, viruses, and protozoa (*Giardia* and *Cryptosporidium*), and for disinfection by-products (DBPs), including trihalomethanes (THMs) and haloacetic acids (HAAs). This rule is intended to provide EPA with information on the occurrence in drinking water of microbial pathogens and DBPs. In addition, EPA collected engineering data on how PWSs currently control such contaminants as part of the ICR.

Under the ICR, PWSs were required to monitor source and treated water for the designated contaminants for a period of 18 months. The 18-month monitoring period started in July 1997. PWSs were required to conduct finished water monitoring at any treatment plant at which it detected, during the first 12 months of monitoring, 10 or more *Giardia* cysts, 10 or more *Cryptosporidium* oocysts, or one or more total culturable viruses per liter of water. The PWSs were to analyze finished water samples for the same organisms analyzed in source water until 18 months of source water microbial monitoring were completed. The data were placed in the ICR Federal Database, available to the public at the following Internet address: http://www.epa.gov/safewater/icr.html.

Finally, consistent with the emphasis on source water protection, a rule to control public health risk from contaminated ground water was included under the SDWA amendments of 1996. An informal draft of the Ground Water Rule (GWR) preamble was posted on the Internet in February 1999. The proposed GWR was published in May, 2000, for public comment (EPA 2000c). This rule specifies the appropriate use of disinfection in ground water and addresses other components of ground water systems to assure public health protection. The GWR establishes multiple barriers to protect against bacteria and viruses in drinking water from ground water sources and will establish a targeted strategy to identify ground water systems at high risk for fecal contamination. The final GWR was scheduled to be issued in November of 2000, but has not yet been promulgated.

The proposed GWR provides several requirements to assure public health protection. These are

- Sanitary surveys to be conducted by the state and identification of significant deficiencies (every 3 years for community water systems, 5 years for non-community water systems; this is consistent with the IESWTR).
- Hydrogeologic sensitivity assessments for undisinfected systems.
- Source water microbial monitoring by systems that do not disinfect and draw from hydrogeologically sensitive aquifers or have detected fecal indicators within the distribution system.
- Corrective action by any system with significant deficiencies or positive microbial samples indicating fecal contamination.
- Compliance testing for systems which disinfect to ensure that they reliably achieve 4-log (99.99%) inactivation or removal of viruses.

Full details of these requirements will be found in the final rule when published.

Threats to Source Water Quality

Two major threats to source water quality with respect to DBP control and microbial protection are natural organic matter and microbial pathogens. The impacts, sources, and challenges to the management of the former are discussed below. The remaining portions of this chapter will address microbial contamination in more detail, in particular contamination by the pathogens *Giardia* and *Cryptosporidium*.

Natural Organic Matter and DBPs

DBPs occur due to the reaction of disinfectants with naturally occurring organic matter (NOM) that is present in all surface waters. Under the Stage 1 Disinfectants/Disinfection By-Product Rule promulgated under the 1996 amendments to the SDWA, water utilities must reduce NOM concentrations, expressed as total organic carbon (TOC), in their raw water to certain specified levels before chlorine is applied for disinfection (Hoehn et al. 1994). Minimum TOC removal requirements vary according to the source water TOC concentration and alkalinity, but range between 20–50% (Hoehn et al. 1994). Since the type and extent of required prechlorination treatment and the ability of a utility to meet the maximum contaminant levels for THMs are dictated by the quality of the raw water, attention has recently focused on understanding, characterizing, and controlling the sources of NOM (Hoehn et al. 1994; Stepczuk et al. 1998; Krasner et al. 1996; Minear and Amy 1996).

Sources of NOM in receiving waters can be broadly categorized as either allochthonous (originating outside the receiving water) or autochthonous sources (originating within the receiving water). Examples of the former include watershed sources such as soils, leaves, and plant remains that are transported to the receiving water by runoff or by tributaries, while autochthonous sources include algal matter, aquatic animals, and bacteria (Cooke and Carlson 1989; Cooke et al. 1988; Hoehn et al. 1994).

The relative importance of NOM sources to the total TOC and THM concentration will vary between receiving waters; Hoehn et al. (1994) provide several examples for a variety of watersheds and receiving waters. Past research has indicated that algae are as potent as humic and fulvic acids from allochthonous sources (Graham et al. 1998; Hoehn et al. 1994) and suggests that, for eutrophic water bodies subject to high nutrient loading from their watersheds, algae is likely to be the greatest source of DBP precursors during the growing season (i.e., spring to fall). Recent modeling efforts by New York City's Department of Environmental Protection for their Cannonsville Reservoir demonstrates the need for nutrient loading to be considered in the management control of THMs for eutrophic reservoirs (Stepczuk et al. 1998).

Anthropogenic loadings of nutrients into our nation's atmosphere and aquatic and terrestrial ecosystems have increased dramatically within the past few decades. Significant watershed loadings are associated with both point and nonpoint sources. Examples of the former include municipal point sources such as sewage treatment facilities offering secondary treatment that characteristically provide minimal nutrient removal; storm water that is enriched from the wet and dry atmospheric deposition of nutrients; combined sewers that discharge nutrient-enriched sanitary sewage and rainwater; industrial discharges; and particulate nutrients associated with runoff from construction sites. Nonpoint or diffuse sources that can be locally important include runoff from overfertilized agricultural lands; animal pastures and waste lagoons; storm water runoff from unsewered communities; septic tank and landfill leachate; particulate nutrients from sediment erosion; atmospheric deposition from mobile sources (e.g., automobiles), power facilities, and confined animal-feeding operations; and nitrogen emissions from receiving waters and terrestrial ecosystems.

Challenges to managing the risk posed by nutrients include the determination of which nutrient to control and by how much; the relative importance of sources (i.e., the relative bioavailability of a source's nutrient load); how the relative importance and abundance of these sources vary spatially and seasonally; and the determination of where and when controls are most needed. Since it is typically the dissolved form of nutrients that are most bioavailable (i.e., most capable of fueling eutrophication), many traditional point and nonpoint source pollution controls that are aimed at removing solids and solids-associated pollutants may be minimally effective at controlling nutrients. In addition, many pollutant controls that remove selected pollutants (e.g., solids, metals) may inadvertently fuel eutrophication through the removal of non-nutrient growth factors (e.g., reduced turbidity removing light transmission limitations). Prior to the successful management of nutrients from both point and nonpoint sources, information is required on the relative importance (i.e., bioavailability) of nutrient sources; when (i.e., which season) controls need to be most effective to prevent ecosystem overfertilization; where in the watershed should controls be placed to maximize the cost-effective control; which pollution controls, best management practices (BMPs), and pollution prevention techniques are most effective at removing the bioavailable forms of nutrients during the critical periods when these loads make their maximum contribution to overfertilization; and the costs and cost effectiveness of these controls, practices, and techniques.

Protocol presently exists for determining numeric nutrient loading targets for a given waterbody (USEPA 1999c); however, the process is not a straightforward one. Research is currently planned that will determine which nutrient(s) to control and by how much for the nation's ecoregions (Garber et al. 1999). Once the nutrient(s) that limit eutrophication have been determined; numeric targets defined; continuous, episodic, and seasonal inputs of natural and anthropogenic sources characterized; and cycling processes identified and their relative importance understood, managers can develop waste load/load allocations and a management plan aimed at achieving the desired reductions for the identified sources. Management options for the control of nutrient sources include point-sources controls (e.g., upgrades at water pollution control plants, emissions controls, etc.); the use of structural and nonstructural BMPs for the control or treatment of nonpoint and diffuse sources; land use controls aimed at decreas-

ing population density, protecting vulnerable areas, or maintaining the assimilation capacity of natural ecosystems; and the restoration of ecosystems capable of intercepting and assimilating nitrogen loads (e.g., riparian zones, forests, or wetlands).

Although BMPs are often employed to treat nonpoint sources of watershed pollutants, including nutrients, significant uncertainty is associated with their ability to control this stressor with removals ranging from 10–90% for some of the more common structural BMPs (Griffen 1993). For this reason, nutrient controls are often targeted at point sources where less uncertainty is associated with both expected removals and costs. Non-site specific factors that may influence the effectiveness of BMPs includes their age, capacity, maintenance, and design specifications. Watershed-specific characteristics that can influence effectiveness include soil characteristics; land use; land cover; climate; site location relative to receiving waters; soil processes and ground water hydrology that can influence pollutant infiltration, decomposition, adsorption, and transport; and biogeochemical processes that may differ between drainage basins (Fisher et al. 1992). When many BMPs are applied to different locations within a watershed, it is still more difficult to predict their integrated effects, and few studies have examined BMP effectiveness for nutrient control on a watershed scale (Edwards et al. 1997; Griffin 1995).

In watersheds where surface waters have been degraded by excessive nutrient inputs, land-use controls are often recommended as a means by which to limit future point and nonpoint nutrient inputs (Minei and Dawydiak 1997). Common controls include the purchasing of farmland development rights; the conservation of forests, wetlands, and riparian lands; and changes in zoning. Where available, watershed models calibrated to actual data or regional or national estimates are often used to predict the pollution potential of various land-use scenarios (Houlahan et al. 1992; Preston 1996; Corbett et al. 1997; Valiela et al. 1997). However, as with BMPs, there may be considerable uncertainty associated with these "alternative futures analyses," in particular where models rely on national or nonlocal estimates of export coefficients.

Restoration of natural features (e.g., riparian forests and wetlands) are often part of management plans aimed at controlling the transport of nutrients to receiving waters. However, the effectiveness of these features at capturing nutrients from upland land uses can be influenced by a number of factors including the magnitude of loadings relative to ecosystem structure (Hopkinson 1992); the relative distribution of natural ecosystems, e.g., uphill versus downhill (Correll et al. 1992); and the infiltration or contact between ground water and root systems (Peterjohn and Correll 1986).

Finally, as the focus of controls shift from point to nonpoint management, the behavior of urban and suburban private land owners may often determine the success of nonpoint and diffuse source control efforts. Although there is recent awareness that economic and social considerations play an important role in the success of nutrient management efforts, few studies have evaluated the role of values, knowledge, income, or other circumstances in an individual's nutrient use and disposal, or the effectiveness of education and economic or other incentives aimed at reducing nutrient loads.

Pathogen Contamination

The potential sources of pathogens in source water are many and varied including nonpoint source runoff, discharges from treated and untreated sewage, and combined sewer overflows. From a waterborne outbreak and public health viewpoint, both *Giardia* and *Cryptosporidium* are of primary concern.

Cryptosporidium is ubiquitous in the environment. Runoff from unprotected watersheds and treated and untreated sewage discharges transports these microorganisms to water bodies used as intake sites for drinking water supplies. Oocysts resist inactivation by commonly used disinfection practices and temperature extremes (Fayer 1994; Fayer and Nerad 1996). As indicated above, *Cryptosporidium* in

source water, particularly source water serving unfiltered surface water systems, requires special attention mandated by EPA's IESWTR (USEPA 1998).

In the U.S., *Giardia* is the most commonly identified pathogen in waterborne disease outbreaks (LADWP 1996). Contamination of a water supply by *Giardia* can occur in two ways: by the activity of animals, particularly beavers, in a watershed or by the introduction of sewage into the water supply.

For many years, detection and enumeration methods for microbial agents in water focused largely on sanitary indicator bacteria, primarily the total and fecal coliforms, *E. coli* and fecal enterococci. Bacterial pathogens such as *Salmonella*, *Shigella*, *E. coli* 0157:H7, and *Campylobacter* have received some attention due to waterborne illness outbreaks. However, other bacteria, viruses, or protozoan pathogens received very little attention until waterborne outbreaks caused by them were documented.

The occurrence of waterborne disease outbreaks has been the key driver of sanitary microbiology research throughout the past 100 years in the U.S. The most recent waterborne pathogens to arrive on the scene have been the pathogenic protozoa, *Giardia lamblia* and *Cryptosporidium parvum*. Contamination of the soil and aquatic environments occurs through shedding of *Giardia* cysts and *Cryptosporidium* oocysts by infected animals, including man. Control of the occurrence of these pathogens in watersheds and their surface waters will be difficult since many animals have been shown to be infected by these organisms, and human sewage contains sizeable concentrations of cysts and oocysts depending on the level of infection in the community. There may be other waterborne protozoan pathogens to be concerned about as indicated by the research being stimulated by the Contaminant Candidate List, finalized in 1998 (63 FR 10274) by the EPA Office of Ground Water and Drinking Water (USEPA 1998b). This section presents background information on waterborne outbreaks due to *Giardia* and *Cryptosporidium* and on the occurrence of these organisms in surface waters used as drinking water sources, storm water run-off, sewage, and combined storm water-sewage overflows (CSOs).

Giardiasis outbreaks were gradually recognized during the period from 1961–1980 (Craun 1986). Diagnosis was by fecal examination of patients, and there was no suitable method for detection of the cysts in environmental water samples. The first identified cryptosporidiosis outbreak occurred in the United Kingdom (U.K.) in 1983, while the first U.S. outbreak of cryptosporidiosis occurred in Braun Station, TX, in 1984 (Lisle and Rose 1995). *Giardia* and *Cryptosporidium* were both formerly thought to be harmless commensals, and it took some time for sufficient information to be developed that showed them to be disease agents. Since 1984, there have been numerous outbreaks of waterborne cryptosporidiosis, including the massive Milwaukee, WI, outbreak in 1993 that affected 403,000 people. In addition, *Giardia* continues to be one of the most frequently identified etiologic agents of gastrointestinal illness due to contaminated drinking water.

The number of waterborne giardiasis and cryptosporidiosis outbreaks that have occurred in the U.S. since 1960 are shown in Table 4-1. The outbreak data in Table 4-1 include both drinking water and recreational water outbreaks. *Cryptosporidium* oocysts, although much smaller (4–6 μm) than *Giardia* cysts (10–14 μm), behave much the same as *Giardia* cysts with regard to physical removal processes in drinking water treatment, but are much more resistant to chemical disinfection than are *Giardia* cysts. However, to assure maximum removal of *Cryptosporidium* oocysts during water treatment, the physical processes must be optimized and consistently operated. Cysts and oocysts can be detected in a sample using the same methodology, although the detection methodology needs much improvement. The number of reported *Giardia* outbreaks tended to increase from year to year once it was acknowledged as a waterborne pathogen. With *Cryptosporidium*, no such trend has been evident, but this may be due to the lack of a good detection method or to other, as yet poorly understood, factors including environmental survival of oocysts, viable and noninfective versus viable and infective oocysts, a high rate of inapparent infections, and infective dose variation due to both the strain of *Cryptosporidium parvum* and individual host susceptibility.

Table 4-1. Summary of U.S. Drinking Water and Recreational Waterborne Disease Outbreaks Due to *Giardia* and *Cryptosporidium*

Date	Parasite	No. Outbreaks	No. Cases	References
1961–1965	*Giardia*	1	123	Craun et al. 1986
1966–1970	*Giardia*	2	53	Craun et al. 1986
1971–1975	*Giardia*	13	5,136	Craun et al. 1986
1976–1980	*Giardia*	26	14,416	Craun et al. 1986
1981–1988	*Giardia*	120	573	Herwaldt et al. 1991
	Cryptosporidium	2	14,966	Lisle and Rose 1995
1989–1990	*Giardia*	7	697	Herwaldt et al. 1991
	Cryptosporidium	0	0	Herwaldt et al. 1991
1991–1992	*Giardia*	8	157	Moore et al. 1994
	Cryptosporidium	5	3,526	Moore et al. 1994
1993–1994	*Giardia*	9	526	Kramer et al. 1996
	Cryptosporidium	9	403,930	Kramer et al. 1996
1995–1996	*Giardia*	3	1,536	Levy et al. 1998
	Cryptosporidium	6	8,572	Levy et al. 1998

Several factors likely account for the increases in the number of reported waterborne outbreaks of both giardiasis and cryptosporidiosis, including (1) recognition that *Giardia lamblia* and *Cryptosporidium parvum* are human pathogens; (2) recognition as waterborne pathogens (first recognized waterborne giardiasis outbreak, 1964–65, first recognized cryptosporidiosis outbreak, 1983 in the U.K. and 1984 in the U.S.); (3) improved detection methodology; and (4) improved surveillance and reporting. Good background articles on *Giardia* and *Cryptosporidium* in water were written by Lin (1985) and Rose (1988), respectively. An extensive review of *Cryptosporidium* spp. and cryptosporidiosis in animals was published by Fayer and Ungar (1986) and Fayer (1997). Marshall et al. (1997) published a review of waterborne protozoan pathogens that includes *Cryptosporidum parvum*, *Giardia lamblia*, and six other protozoans as well as a section on water quality protozoan testing and monitoring. Craun et al. (1998) reviewed 35 waterborne cryptosporidiosis outbreaks associated with contaminated drinking water and recreational activities, provided recommendations for prevention of such outbreaks, and assessed the need for epidemiological data.

Pathogens in the Environment and in Wet Weather Flow

Cysts and oocysts are common in surface water, and the concentration appears to vary with watershed use characteristics (Hansen and Ongerth 1991). It has been established that oocysts are found in most surface waters as shown in Table 4-2. Hansen and Ongerth (1991) found oocysts in 34 of 35 river samples, using a method with a detection limit ranging from 0.04 to 0.14 oocysts per liter and a recovery efficiency of 18.6 to 34.3%. The watersheds examined had a variety of nonurban land uses. In a study conducted on the Allegheny River, *Cryptosporidium* was detected in 50% or more of all samples collected (NRCS 1997). Samples were collected over a 3½ year period, with a recovery efficiency of just 25%. Roughly 22% of the samples collected in the New York City watershed showed *Cryptosporidium* oocysts, with a slightly greater fraction showing *Giardia* cysts (Stern 1996). The background concentration in one drinking water reservoir was estimated at 0.36 oocyst/100 L (Stewart et al. 1998).

Research to determine correlations between *Giardia* and *Cryptosporidium* and other parameters has been inconclusive. LeChevallier et al. (1991a) measured *Giardia* and *Cryptosporidium* in the source waters of 66 surface water treatment plants in the U.S. and Canada. They identified oocysts in 87% of

Table 4-2. Occurrence of *Giardia* Cysts and *Cryptosporidium* Oocysts in Surface Waters

Water Type	No. of Samples	% Samples Positive *Giardia*	% Samples Positive *Crypto.*	Conc. Range Per L (GM)# *Giardia*	Conc. Range Per L (GM)# *Crypto.*	Reference
Surface	51	39	39	–	–	Barthe and Brassard 1996
Rivers/lakes	181	15	51	<0.01–1.4 (0.03)*	<0.01–44 (0.43)*	Rose et al. 1991
Allegheny R.	24	63	63	0–4.2 (0.34)	0–22.3 (0.31)	States et al. 1997
Youghiogheny R.	24	54	63	0–5.3 (1.2)	0–14.7 (0.58)	States et al. 1997
Stream, dairy farm	24	54	82	0–15.7 (0.82)	0–11.05 (0.42)	States et al. 1997
River diversion	19	21	50	0–6.25 (0.22)	0–240 (1.09)	Rose et al. 1988
Lake outlet	20	40	50	0–2.22 (0.08)	0–22 (0.58)	Rose et al. 1988
Stream/river	11	–	77.6	–	2–112 (25.1)*	Ongerth and Stibbs 1987
Surface	107	–	77	–	0.04–18 (0.91)	Rose 1988
Reservoir inlet	60	13.3	5	0.007–0.24 (0.19)	0.007–0.024 (0.012)	LeChevallier et al. 1997
Reservoir outlet	60	15	11.7	0.012–1.07 (0.061)	0.017–0.31 (0.081)	LeChevallier et al. 1991a
Surface water	85	81	87	0.04–66 (2.77)	0.07–484 (270)	LeChevallier et al. 1991a
River/stream canal water	6	–	ng^	–	0.8–5,800 (ng)	Madore et al. 1987
Raw source waters	262	45	51.5	0.02–43.8 (2.0)	0.065–65.1 (2.4)	LeChevallier and Norton 1995

GM = geometric mean
* = arithmetic mean
^ ng = not given

sampled surface waters, reporting higher densities in waters receiving industrial or sewage effluents and also significant correlations between *Giardia* and *Cryptosporidium* concentrations with turbidity and fecal coliform concentrations. *Giardia* and *Cryptosporidium* concentrations reported in 39% of the surface waters sampled in Canada showed no correlation with either total or fecal coliform concentrations, heterotrophic plate count, pH, temperature, turbidity, or dissolved organic carbon (Barthe and Brassard 1996). One factor to consider in explaining this inconsistency is that reported oocyst concentrations included both viable and nonviable organisms (LeChevallier et al. 1991b).

A national study detected *Giardia* spp. in 81% of source water samples from 66 surface water treatment plants in 14 states and one Canadian province. *Cryptosporidium* spp. were found in 87% of the raw water locations. Higher cyst and oocyst densities were associated with source waters receiving industrial or sewage effluents. Significant correlations were found between *Giardia* and *Cryptosporidium* densities, turbidity, and total and fecal coliform levels. Statistical modeling suggests that cyst and oocyst densities could be predicted on the basis of watershed and water quality characteristics (LeChevallier et al. 1991a).

Concentrations of *Cryptosporidium* and *Giardia* in the Delaware River, a drinking water source for several municipalities including New York, NY, Philadelphia, PA, and Trenton, NJ, increased after rainfall events. The increased *Cryptosporidium* and *Giardia* concentrations correlated with increased coliphage, total coliform, fecal coliform, *E. coli*, and *C. perfringens* concentrations. The increase was attributed to transport through surface runoff, resuspended storm drain, and river bottom sediments (Atherholt et al. 1998).

Giardia cysts were found in 94 (43%) of the 222 samples collected over a nine-month period from 17 sampling stations from three pristine rivers in the Pacific Northwest (Ongerth 1989). No statistically supportable seasonal variations were found. *Giardia* cysts were continuously present, though at low concentrations, even in relatively pristine rivers (Rose et al. 1991; Rose 1997).

Giardia cysts and *Cryptosporidium* oocysts have been found at low levels in ground waters and springs, as summarized in Table 4-3. In general, contamination of well waters appears more likely for *Cryptosporidium* oocysts than for *Giardia* cysts, but well depth, construction, and state of repair will strongly influence the possibility of contamination. Regardless of the type of well or spring, cyst and oocyst concentrations were usually found to be low.

Table 4-3. *Giardia* and *Cryptosporidium* in Springs and Ground Waters

Source	No. Samples or Sites	% Samples Pos.		Range Cyst/Oocysts/100L		References
		Crypto.	*Giardia*	*Giardia*	*Crypto.*	
Well waters	20	0	0	–	–	Barthe and Brassard 1996
Ground waters	18	–	5.5	<0.25	(0.3)*	Rose et al. 1991
Spring, pristine	7	0	57.1	<0.25	<0.25–13(4)	Rose et al. 1991
Vertical wells	149	1	5	–	–	Hancock et al. 1998
Springs	35	14	20	–	–	Hancock et al. 1998
Infiltration galleries	4	25	50	–	–	Hancock et al. 1998
Horizontal wells	11	36	45	–	–	Hancock et al. 1998
Total sites	199	12	12	0.1–120(8)	0.2–45(5)	Hancock et al. 1998
Deep well, pristine	288	–	0	–	–	Benton et al. 1991
Well, coliform positive	138	–	5.8	–	4–92(23)	Badenoch et al. 1990

– = data not given
* = arithmetic mean

Sources of Oocysts

Giardia cysts and *Cryptosporidium* oocysts are found at significant levels in domestic raw sewage, treated sewage effluents, and CSOs. Table 4-4 summarizes data on *Giardia* and *Cryptosporidium* cysts/oocysts in sewage and CSOs. Source identification and characterization play an important role in determining potential control measures. For example, SWP measures for oocysts from human sewage versus animal sources will be different. Even if the cysts and oocysts are known to be from human sewage, there may still be considerable differences in control options available, depending on whether the cysts and oocysts were discharged due to faulty septic systems, wastewater treatment plant effluent, treatment plant bypass, sanitary sewer overflows (SSOs), CSOs, or storm water. Similarly, significant differences in options occur if the animal source of the oocysts is from a dairy farm, cattle ranch, a concentrated feed, or wild animal populations.

Table 4-4. *Giardia* and *Cryptosporidium* Cysts/Oocysts in Sewage and Combined Sewer Overflows

Source	No. of Samples	% Samples Positive		Range Cyst/Oocysts per L(GM)[a]		Reference
		Giardia	*Crypto.*	*Giardia*	*Crypto.*	
Raw sewage	29	100	0.03	130–7900 (1,500)[a]	0–28(ng)	Hirata and Hashimoto 1997
Primary effluent	37	100	–[b]	150–6,600 (1,100)[a]	–	Hirata and Hashimoto 1997
Final effluent	33	82	0	4–130 (14)	–	Hirata and Hashimoto 1997
Sewage[d] influent	24	100	100	200–3,200 (ng)	–	Casson et al. 1990
Return act. sludge	8	100	100	200–900 (ng)	–	Casson et al. 1990
STP trick. filter	8	100	100	4–44 (11)	–	Casson et al. 1990
Raw sewage	–	–	–	11–397 (ng)		Sykora et al. 1987
STP effluents	–	–	–	0.01–13.5 (ng)	–	Sykora et al. 1987
Raw sewage	3–36	100	14	26–3, 022 (ng)	0–74(ng)	Roach et al. 1993
STP effluents	–	–	–	2–3, 511 (ng)	0–333(ng)	Roach et al. 1993
Sewage effluent	15	80	27	0–4, 614 (42)	0–4, 927 (43.2)	States et al. 1996
Raw sewage	4	–	100	–	850–13,700 (51.8)[b]	Madore et al. 1987
Treated sewage	9	–	–	–	140–3,960 (1,060)[b]	Madore et al. 1987
Combined overflows, upper, in stream, d.w.[c]	6	100	100	<0.13–0.66 (36)	0.05–0.53 (0.18)	Gibson et al. 1998
Lower, in stream, d.w.	6	100	100	0.21–66 (3.43)	<0.33–1.05 (0.78)	Gibson et al. 1998
Upper, in stream, w.w.[c]	3	100	67	0.67–2.88 (1.15)	<0.39–0.72 (0.70)	Gibson et al. 1998
Lower, in stream, w.w.	3	100	100	4.29–75 (26.5)	4.29–1.77 (7.5)	Gibson et al. 1998
End of pipe	11	100	100	90–2,830 (354)	2.5–400 (60.4)	Gibson et al. 1998
CSO	5	80	–	37–1,140 (287)[d]	8.8–30 (20.1)[d]	States et al. 1997

[a] geometric mean number of cysts/oocysts/L; ng = not given
[b] arithmetic mean
[c] d.w. = dry weather; w.w. = wet weather
[d] 8-hour composite samples

There is little information on septic tanks as a potential source of *Cryptosporidium*. Septic tanks that function poorly are possible sources of oocysts and need to be addressed for public health reasons, including *Cryptosporidium*. The New York City Department of Environmental Protection is conducting a study on the transport of oocysts from functioning septic systems, and a report on its findings was to be available in December 1999 (USEPA 1997).

A variety of mammals, particularly young ruminants, are sources of *Giardia* cysts and *Cryptosporidium parvum* oocysts in the environment. Table 4-5 presents some information on concentrations of *Giardia* cysts and *C. parvum* oocysts in the feces from humans and some animals.

Table 4-5. Some Human and Animal Sources of *Giardia* Cysts and *Cryptosporidium Oocysts*

Source	No. Samples	% Samples Pos		Cysts	Oocysts	Reference
		Giardia	*Crypto.*	*Giardia*	*Crypto.*	
Human						
Infected	–	–	–	3×10^3/ person/day	–	Erlandsen and Meyer 1984
AIDS patient, infected	–	–	–	–	6×10^6– 1.2×10^{10}	Goodgame et al. 1993
Agricultural						
Calves/lambs	–	–	–	–	10^{10}/day to 14 days	Current and Garcia 1991
Calves, infected	–	–	–	–	10^3/g, 5–15 Kg feces per day	Breach et al. 1994
Cow, infected	–	–	–	–	10^4/g, 25–30 Kg feces per day	Breach et al. 1994
Cattle, infected	108	–	26.8	–	–	Quilez et al. 1996
Swine, infected	90	–	34.4	–	–	Quilez et al. 1996
Parks/ Recreational						
Beaver	–	–	–	–	–	Erlandsen and Meyer 1984
Muskrat	–	–	–	–	–	Erlandsen and Meyer 1984
Canada geese	9*	100	77.7	75–786/ g feces	67–686/ g feces	Graczyk et al. 1998

*pooled sample

Eighty species of mammals have been shown to shed *C. parvum* oocysts (Barry et al. 1998). Most measurements have been completed on domestic animals, with little information available regarding the shedding of *C. parvum* by wildlife. In addition to humans, among the domestic and wild animals found to be hosts for *C. parvum* are cattle (Atwill et al. 1998; Xiao and Herd 1994; Kuczynska and Shelton 1999; Garber et al. 1994), sheep, goats, deer, water buffalo, pigs (Atwill et al.1997), horses (Forde et al. 1997; Haas and Rose 1994; Johnson et al. 1997), rabbits, opossum, rodents (rats, Webster and McDonald 1995; mice, Klesius et al. 1986; Bajer et al. 1997), beaver and muskrats (Bajer et al. 1997), migratory water fowl, and primates (Graczyk et al. 1998b).

Oocyst Survival

The ability of oocysts to survive rather harsh environments (e.g., low temperatures, typical drinking water chlorination) enhances their chances of successfully migrating to a treatment plant intake and through the treatment process. Understanding conditions that oocysts can and cannot tolerate can be instrumental in devising effective controls or in estimating when high levels of oocyst survival will occur in source waters (Walker 1998; Graczyk et al. 1998a; Fayer and Nerad 1996).

Fayer and Nerad (1996) have shown that, although freezing at very low temperatures (–70°C) inactivated oocysts, freezing at higher temperatures (–10, –15, and –20°C) allowed oocysts to retain some level of infectivity. Oocysts frozen at –20°C for five hours or less remained infective. Oocysts frozen at –10°C for 168 hours or less, as well as those frozen at –15°C for 24 hours or less, also remained infective. Although freezing temperatures are detrimental to oocyst survival, this study suggests that, when the surface does not reach low temperatures (below –10°C) for prolonged periods of time, some infective oocysts may survive for extended periods.

Jenkins et al. (1999) performed field studies of oocysts exposed to the environment of calf manure piles and the surface of a field soil. Results of this study indicated that exposure to both manure and soil environments significantly increased rates of oocyst inactivation compared to controls. Exposure to freeze-thaw cycles in soil were particularly deleterious to the oocysts. They concluded from their study that spreading manure contaminated with oocysts on snow, in an absence of freeze-thaw cycles, may contribute to sustained oocyst survival and increase the risk of surface water contamination during spring melt and runoff.

Fayer (1994) examined the effect of high temperatures on oocyst infectivity. Oocysts were rendered noninfective within one minute upon reaching a temperature of 72.4°C or higher. Oocysts held at 64.2°C or higher for 2 minutes also lost their infectivity. This study was conducted on oocysts in distilled water. It is possible that temperatures needed to inactivate oocysts on land or in compost may vary. Jenkins et al. (1999) suggest that oocyst infectivity may be significantly reduced within 70 days in manure piles with temperatures between 35 and 50°C.

Jenkins et al. (1998) reported that low concentrations of ammonia associated with a barnyard environment inactivated oocysts and can inactivate a viable population in days. They also demonstrated that the pH associated with the various levels of ammonia tested, pH 9 to 11, was not a factor associated with their inactivation.

Results of one study (Chauret et al. 1998) examining the role of biological antagonism in the inactivation of oocysts suggest that biological antagonism may be a primary factor affecting oocyst survival in natural waters. However, this process of natural interactions between organisms appears to be site specific.

Measuring and Monitoring Pathogens in Source Waters

Except for the use of immunological and molecular methods (genetic probes, polymerase chain reaction [PCR], ribotyping) for specific identification of isolates of pathogenic bacteria, cultural methods for the detection, enumeration, and identification of waterborne pathogenic bacteria have not changed significantly over the past three decades. Although somewhat dated, Singh and McFeters (1992) reviewed detection methods for pathogens in water. A comprehensive source of information on environmental microbiology and microbial detection methods may be found in Hurst et al. (1997).

Although most water monitoring involves searching for indicators of fecal pollution, monitoring water for the presence of pathogens is necessary under special circumstances, such as during and after water-

borne outbreaks, when dealing with a water source with a history of contamination, or where wastewater reclamation is involved. The low densities of pathogens usually found in water require that large volumes of water must be examined. For viruses and parasites, this is usually done by filtering a large volume (10–1000 liters [2.6–264 gal]) of water through a filter cartridge to concentrate the target organisms. Sometimes volumes of one liter or more are concentrated by centrifugation or a combination of filtration and centrifugation. The use of large volume samples limits the number of samples that can be examined and increases the costs of testing. Overall, the costs for analysis of water samples for specific bacterial pathogens and for enteric viruses and protozoan pathogens are quite high, with those for viruses and protozoan cysts and oocysts being much higher than for bacteria. Given the limitations of detection and enumeration methodologies and their complexities, a negative result for finding a specific pathogen does not mean that no target pathogens were present, only that none were detected at the detection limit of that method.

Giardia *and* Cryptosporidium

The methods currently in use for *Giardia* and *Cryptosporidium* detection in water have been developed since the early 1980s. Since 1992, with the development of a series of regulations (D/DBP Rule, IESWTR, and the ICR), water utilities have been in need of water quality testing laboratories, either in-house or via contract, with the capability of analyzing for *Giardia* cysts and *Cryptosporidium* oocysts in finished drinking water and in source waters. The method of choice in the U.S. for detection of *Giardia* and *Cryptosporidium* in source waters was the proposed American Society for Testing and Materials (ASTM) analytical procedure. The method is technically complex, labor intensive, time consuming, and requires good laboratory quality-control procedures to provide maximum recoveries. The method performance is also affected by variations in sample collection, water quality (turbidity) and analyst training, experience, and competency.

The EPA method used for the ICR (USEPA 1996) differs from the ASTM method by requiring filtration of 100L (26.4 gal) of raw water or 1000 L (264 gal) of finished water and the use of Hoffman modulation or differential interference contrast (DIC) optics instead of phase-contrast optics for confirmation of morphological characteristics of the presumptive cysts and oocysts. Because of method performance issues, modifications to the EPA method resulted in the development of EPA Methods 1622 and 1623 (USEPA 1999a, 1999b), respectively. Each method uses sample concentration by filtration, combined with immunomagnetic separation and fluorescent antibody staining for recovery and enumeration of cysts and oocysts. Method 1622 is a stand-alone method for *Cryptosporidium*, while Method 1623 is for simultaneous detection of *Giardia* and *Cryptosporidium*.

Enteric Viruses

Viruses are present in very low numbers in most environmental waters. Therefore, methods for the detection of enteric viruses in water, as for the protozoan pathogens, also require concentration of the viruses from a large volume water sample following a protocol involving filtration and centrifugation, recovery of the viruses from the filtration medium, and assay of the concentrated sample for viruses by inoculation into a mammalian cell culture line. The methods published in the USEPA Manual of Methods for Virology (Berg et al. 1983) and in Standard Methods for the Examination of Water and Wastewaters (APHA 1999) were probably the most commonly used prior to the ICR method. However, the ICR virus monitoring protocol, developed by the EPA and modified by consensus agreements from the scientific community (USEPA 1996), represents the methodology most used during the 1990s for detecting enteric viruses in water.

Protecting Source Waters

This section discusses protection of source water from microbial pathogens found in treated sanitary sewage and wet weather flows (i.e., SSOs, CSOs, and storm water runoff).

Separate Sanitary Sewage Systems

Separate sewage systems require a dedicated infrastructure to carry waste to the treatment plant. Typically, these systems are largely gravity flow-augmented by pumping stations if needed. The system is designed to meet specified flow quantities, and balances flow from all influents with the treatment plant throughput capacity. When the demand exceeds the flow capacity of the system, a surcharge, or SSO, occurs. Under surcharge conditions, the system discharges through alternate escape routes, often backing up into residences or streets, and eventually winding up in receiving waters. Separate sewage systems are seldom leakproof. Connections between pipe sections along the length of the conveyance system are not completely sealed. The connections and the privately owned laterals offer opportunities for waste to escape and for subsurface water to infiltrate the system. Inflow and infiltration can be substantial during rain events, decreasing the flow capacity available for the wastewater. Table 4-6 presents representative data on the type and number of microorganisms found in untreated wastewater (Metcalf & Eddy Inc. 1991). The table reports densities of both indicator and pathogenic microorganisms.

Table 4-6. Types and Numbers of Microorganisms Typically Found in Untreated Domestic Wastewater

Organism	Concentration (number/mL)
Total coliform	10^5–10^6
Fecal coliform	10^4–10^5
Fecal streptococci	10^3–10^4
Enterococci	10^2–10^3
Shigella	Present[a]
Salmonella	10^0–10^2
Pseudomonas aeroginosa	10^1–10^2
Clostridium perfringens	10^1–10^3
Mycobacterium tuberculosis	Present[a]
Protozoan cysts	10^1–10^3
Giardia cysts	10^{-1}–10^2
Cryptosporidium cysts	10^{-1}–10^1
Helminth ova	10^{-2}–10^1
Enteric virus	10^1–10^2

[a] Results for these tests are usually reported as positive or negative rather than being quantified.

Concentrations of microorganisms in sewage treatment plant effluent vary depending on the National Pollutant Discharge Elimination System (NPDES) permit issued for the plant (King 1996). The effluent concentrations and level of treatment required are those necessary to achieve receiving water quality standards. Receiving water standards have been established pursuant to the Federal Water Pollution Control Act Amendments of 1972 (Public Law 92-500) to protect beneficial uses of surface waters. One goal is to eliminate pathogens to control transmission of waterborne diseases. In support of this goal, wastewaters that pose a disease risk are disinfected prior to discharge. Generally, NPDES permits require measuring the microbial indicator concentrations in the effluent rather than pathogen concen-

trations. Therefore, treating wastewater so that the permit standard is met does not guarantee an absence of pathogenic microorganisms. Indicators are more representative of some pathogens than others. Olivieri et al. (1977) found that, in raw sanitary sewage, there was a strong positive correlation between the levels of total coliform (TC), fecal coliform (FC), fecal streptococci (FS), and enterococci and the levels of pathogenic bacteria. Only the levels of TC and FC correlated well with the levels of enteric viruses. Metcalf et al. (1995) reported that, on average, virus concentrations of about 50 plaque forming units/liter (PFU/L) can be expected in wastewater treatment plant effluents.

Combined Sewer Systems

The design of combined sewer systems commingles sanitary and storm water flows in a single conveyance system that routes the entire flow to the wastewater treatment plant. This system treats all collected liquid, including storm water and sanitary sewage, before discharge to the receiving water. By using a single conveyance to carry all flows to the treatment plant, this design requires a total pipe length less than that required by separate storm and sanitary sewers.

At construction, the combined sewer system is sized to meet the existing and projected flows of sanitary (dry-weather flow [DWF]) and storm water flows. Design values are available for the sanitary contribution from various types of structures (e.g., hospital, school, or private home). The DWF volume varies over the course of a day, with morning and evening peaks. Weekend flow patterns differ from the work-week flow patterns. Designers base the storm water flow contribution on regional rainfall statistics and the probability of a given storm-induced runoff volume. Whenever storms generate runoff to create combined sewage flow volumes greater than the capacity of the system, the system relieves the pressure by shunting flow to receiving waters, i.e., a CSO occurs. The total flow of overflow events is often expressed as a multiple of the peak DWF. As combined sewer systems age, the number of sanitary users connected to the system increases. The increased sanitary flow volumes deplete capacity formerly used by storm water and increase the frequency of overflows. Similarly, the increased impervious area associated with the new connections increases the storm water runoff, which also consumes conveyance capacity.

EPA's CSO Control Policy (USEPA 1994) limits the number of annual overflows for combined systems and requires disinfection after primary clarification, using the capacity of the publicly owned treatment works, when it is required by local authorities. Systems are being modified to reduce the number of overflows by providing for in-system storage, on-lot storage (Milliken 1996), and disconnecting inputs such as downspouts. Low-impact development is a method currently being evaluated for reducing runoff volumes (Coffman et al. 1996). Its objectives include restoring the site hydrologic's regime to reflect the natural or predevelopment condition and minimizing the generation and off-site transport of pollutants via storm water runoff.

CSO disinfection is practiced to control the discharge of pathogens and indicator microorganisms into receiving waters. Chlorination is the conventional approach to disinfection. Due to concerns about chlorine's effects on aquatic life, alternative technologies are being investigated for CSO disinfection. The New York City Department of Environmental Protection recently completed evaluations of high-rate disinfection technologies (Camp, Dresser, & McKee and Moffa & Associates 1997). Table 4-7 shows the disinfectant dosages associated with achieving effluent microbial indicator concentrations of 1000 colony forming units (cfu)/100 mL using chlorine, ultraviolet (UV) irradiation, ozonation, and chlorine dioxide. Generally, all four technologies were able to provide 3- to 4-log bacterial reductions. UV disinfection was found to provide reduced effectiveness at higher total suspended solids concentrations (>150 mg/L). Chlorine dioxide disinfection requires doses significantly lower than those required for chlorine disinfection, reducing the toxicity impacts on the receiving water aquatic life.

Table 4-7. Estimated Disinfection Dosages to Attain Microbial Indicator Concentrations of 1000 cfu/100 mL (Camp, Dresser & McKee and Moffa & Associates 1997)

	Influent Concentration (cfu/100 mL)	Estimated Dosages			
		Chlorine Dose (mg/L)	UV Dose (mW-s/cm^2)	OzoneDose (mg/L)	Chlorine Dioxide Dose (mg/L)
Total coliform	10^6–10^7	>30[a]	>100[b]	37	>8[c]
Fecal coliform	10^5–10^6	18	50	24	6
E. coli	10^5–10^7	17	35	23	5.5
Enterococcus	10^4–10^6	22	35	12	5.5

[a] Target concentrations not achieved with highest applied dosage, i.e. 30 mg/L.
[b] Target concentrations not achieved with highest applied dosage, i.e. 100 mW-s/cm^2.
[c] Target concentrations not achieved with highest applied dosage, i.e. 8 mg/L.

Particles associated with or occluding microorganisms can reduce the effectiveness of wastewater disinfection by chlorination and by UV irradiation (Parker and Darby 1995). UV irradiation showed lower effectiveness at suspended solids concentrations above 150 mg/L in the New York City studies (Stinson et al. 1998). Understanding the effects of solids content on disinfection effectiveness is necessary for designing treatment systems capable of achieving effluent requirements. Recent EPA research suggests greater disinfection effectiveness is possible by removing solids before UV irradiation and chlorination (Perdek and Borst 2000).

Lijklema et al. (1986) report that CSOs result in increased indicator organism concentrations in the receiving water. They measured TC, FC, *E. coli* FS, somatic coliphages, and F-specific coliphages. Phage concentrations are one to three orders of magnitude smaller than bacterial concentrations. The ratio between the concentrations of different bacterial indicators varies between events, but is generally within 1.5-log units. Ellis and Yu (1995) report that CSOs serve as very effective generators of bacteria and pathogens in urban receiving waters, particularly where available dilution volumes are restricted. A recent EPA investigation showed a thirtyfold increase in FC and enterococci concentrations 28 hours after disinfection by UV light (Wojtenko 1999). Enterococci regrowth after disinfection by chlorine or chlorine dioxide was negligible over the period studied.

Municipal Separate Storm Sewer Systems

The design of separate sewer systems isolates the sanitary and storm water flows. The sanitary sewage flows to the wastewater treatment plant. The second system routes storm water to nearby receiving waters. In this system design, the storm water runoff remains untreated carrying all the associated contaminants, including microorganisms, directly to the receiving water. The system storm water design capacity is based on expected storm runoff volume under the proposed or existing development. Because the system isolates the two flows, the design requires separate conveyance systems with longer total pipe length and long-term maintenance costs than combined systems. With added development, sanitary and storm systems are sometimes connected inappropriately, resulting in sanitary sewage being carried to the receiving water with no treatment.

The presence of microbial indicators and pathogens in storm water has been confirmed. Olivieri et al. (1977) reported high densities of indicator organisms in urban streams in Baltimore, plus the presence of *Pseudomonas aeruginosa*, *Staphylococcus aureus*, *Salmonella*, and enteric viruses. Analyses of storm water in the study area reported the occurrence of pathogenic bacteria and viruses in storm water runoff. The bacterium *P. aeruginosa* was found in all storm water samples taken from six sampling loca-

Table 4-8. Microbiological Concentrations of Storm Water

Contaminant	Concentrations (per 100 mL) in Storm Water	References for Storm Water
Total coliforms	$7–1.8 \times 10^7$	Dutka and Tobin 1978; Dutka and Rybakowski 1978
Fecal coliform	$0.2–1.9 \times 10^6$	Dutka and Tobin 1978; Dutka and Rybakowski 1978
Fecal streptococci	$3–1.4 \times 10^6$	Dutka and Tobin 1978; Dutka and Rybakowski 1978
Enterococci	$1.2 \times 10^2–3.4 \times 10^5$	Gannon and Busse 1989
HPC (#/mL)	$6.94 \times 10^4–4.9 \times 10^5$	Dutka and Tobin 1978
Pseudomonas aeruginosa	$1–1.1 \times 10^7$	Dutka and Tobin 1978; Olivieri et al. 1977
Escherichia coli	$1.2 \times 10^1–4.7 \times 10^3$ $5.7–4.5 \times 10^3$	Gannon and Busse 1989
Salmonella	(MPN/10 L)	Geldreich et al. 1968
Shigella	Not detected	Olivieri et al. 1977
Klebsiella	$4 \times 10^3–1.9 \times 10^5$	Schillinger and Gannon 1985
Enterobacter	Not detected	Dutka and Tobin 1978
Citrobacter	Not detected	Dutka and Tobin 1978
Yersinia enterocolitica	Not detected	NA
Staphylococcus aureus (MPN/100mL)	$1–1.2 \times 10^2$	Olivieri et al. 1977
Legionella	Not detected	NA
Streptococcus	Detected	Geldreich et al. 1968
Viruses (enteric)	Detected	Olivieri et al. 1977
Giardia	–	NA
Cryptosporidium	–	NA
Fungi	$6 \times 102–1.2 \times 10^7$	Dutka and Rybakowski 1978
Parasites–nematodes	Detected	Dutka and Rybakowski 1978
Helminth ova	–	NA

HPC = heterotrophic plate count, NA = none available

tions. *Staphylococcus aureus* and *Salmonella* spp. were found at each of the six sampling locations in a majority of the samples taken. Coxsackievirus B, animal virus, poliovirus, and echovirus were found in storm water samples collected from all six of the sampling locations. Makepeace et al. (1995) summarized concentrations of microbial indicators and pathogens found in storm water runoff reported by others, which is presented as Table 4-8.

Sampling was conducted during the summer of 1985 to evaluate the impacts of discharges from storm drains on bacteriological quality on the Huron River in the Ann Arbor, MI, area during both dry and wet weather periods (Gannon and Busse 1989). Each river water sample was analyzed for FC, FS, *E. coli*, and enterococci. The investigators reported that wet weather bacterial densities were statistically significantly higher than dry weather levels, and downstream densities were statistically significantly higher than upstream densities. The FC/FS ratios for the storm drains were low, suggesting that sources were more animal than human.

A 1999 study to determine the source of unexpectedly high river and stream bacterial contaminations near Nashville showed that FC densities were directly related to the density of housing, population, development, percent impervious area, and apparent domestic animal density. The data also showed

that FC counts were much higher in summer than winter, suggesting a possible seasonal variation. The FC/FS ratios were generally low, suggesting primarily animal sources. Surface runoff samples from more densely populated sewered areas generally showed higher bacterial counts than runoff from less developed areas that utilized septic tanks. The investigators concluded that surface runoff from high density urban areas may be a contributor to high fecal bacteria loadings (Young and Thackston 1999). Consistent with these results are those presented by Mallin (1998), who reported patterns of increasing coliform bacteria concentrations in stream samples with increased watershed development and impervious surface in New Hanover County, NC.

Storm water discharges are regulated in selected communities through the NPDES program (USEPA 2000b). In response to the 1987 amendments to the CWA, EPA developed Phase I of the NPDES Storm Water Program in 1990. Phase I requires NPDES permits for storm water discharges from

- Medium and large municipal separate storm sewer systems (MS4s), generally serving or located in incorporated places or counties with populations of 100,000 or more people.
- Eleven categories of industrial activity, one of which is construction activity that disturbs 5 acres or greater of land.

The Final Rule for Phase II of the NPDES Storm Water Program was signed by the EPA Administrator on October 29, 1999. The Phase II Rule requires NPDES permit coverage for discharges from

- Certain regulated small MS4s (primarily all those located in urbanized areas).
- Construction activity disturbing between 1 and 5 acres of land.

Sediment Resuspension

Pettibone and Irvine (1996) reported levels and sources of indicator bacteria in the Buffalo River, NY, watershed and found that solids present in the water column may offer a vehicle by which bacteria are kept in suspension and transported downstream. Additionally, the sediments provide an environment that promotes microorganism growth and protects them from predators. Sherer et al. (1992) reported longer survival of FC and FS in sediment-laden waters than in the sediment's supernatant and in waters without sediment. When incubated with sediment, FC and fecal Streptococcus half-lives were determined to be from 11 to 30 days and from 9 to 17 days, respectively. These are longer than when they are incubated without sediment.

Best Management Practices

In addition to the installation of sewage treatment and combined overflow systems, there are passive pollution prevention and mitigation techniques called best management practices (BMPs). The techniques vary dramatically in application, ranging from social practices to engineering applications. Even the more heavily engineered solutions combine the practitioner's art with traditional engineering tools and rely on common sense approaches to what should work in a given situation.

BMPs are often categorized according to the degree of structural intensity associated with the practice. Low-structural intensity techniques include public education, emphasizing the consequences of specific actions. Many communities, for example, paint fish on storm sewer catch basins to emphasize the link between potential waste disposal and receiving waters. Similarly, community master planning can incorporate practices intended to prevent contaminant introduction. Mitigation techniques can range from requirements for storm water controls during the development process, including leaving designated areas undisturbed, to housing density controls through zoning. The effectiveness of these pollution prevention techniques is, and is likely to remain, uncertain. Even when installed as part of a remedial approach, it is unlikely that investigations can separate the effects of these approaches. The

temporal scale similarly confounds investigations as source elimination will often require many years of natural flushing and attenuation to become apparent in the receiving waters.

Well-planned and well-executed studies of more structurally intensive approaches are also limited and questions of long-term cost and effectiveness remain. A complete evaluation of a given technique requires mass balances over several seasons to ensure that BMP effectiveness does not simply change with timing, i.e., pollutants temporarily accumulate and are discharged in a later storm event. This phenomenon can be identified in some event-specific and short-term evaluations when effluent concentrations, masses, or both exceed the influent (Kurz 1998). These studies are difficult to complete and depend heavily on flow measurement and sample analysis. The inability to automate analytical processes with data logging sensors makes these evaluations expensive.

Although little well-documented research is available presenting the capabilities to control microorganisms, watershed managers routinely install BMPs for storm water treatment. There is strong suggestive evidence that these installations preserve water quality and can reduce water treatment costs. Kurz (1998) documented pathogen and indicator reductions in sand filtration, wet detention, and alum coagulation treatment systems using simulated storm events. Each system produced significant reductions in TC and FC bacteria, male-specific coliphage 2, and beads (used as a protozoa surrogate) concentrations. Often, effluent samples showed greater concentrations of TC, turbidity, and total suspended solids than influent samples. These increases show the incomplete understanding of the mechanisms, processes, and temporal scales of BMP operation.

In 1999, EPA released fact sheets on the use of sand filters, wetlands, and vegetative swales for managing storm water (USEPA 2000a, 2000b, 2000c). Other BMPs include detention ponds, buffer strips, and infiltration trenches. Sand filters are structurally intensive devices installed primarily to remove particulate and particulate-associated contaminants. Sand beds block the migration of particulates as water passes through the media bed. Some biological activity develops as biofilms develop within the device. Augmenting the media with high organic matter such as peat increases sorption within the filter. Sand filters provide very limited flow modification and therefore provide little protection of streambed or stream banks. Filter sizing is based on predicted runoff volume and is therefore based on the size and infiltration properties of the drainage area. These devices have impermeable bottoms to prevent infiltration to ground water. The filters need to have the filter media replaced periodically depending on loading. Typical replacement periods range from three to five years, with expended filter media suitable for landfill disposal (USEPA 1999a).

Storm water wetlands are incidental, natural, or intentionally constructed areas that are usually flooded. Within these areas, physical, chemical, and biological processes trap or degrade entering contaminants. Intentional use of naturally occurring wetlands to treat storm water runoff may be discouraged or prohibited. Storm water wetlands are divided into subsurface and free water surface systems based on the water flow pattern within the wetland. The selected location must have an adequate water supply and appropriate soil characteristics. Sizing techniques vary and may be state regulated. Common approaches include a designated design storm, fraction of watershed area, and sizing to contain the runoff volume generated by most rain events for the local area. Sources recommend an aspect (length to width) ratio between 1 and less than 10 to reduce internal short circuiting. Wetlands are commonly augmented with ponds (USEPA 1999b). Typical reported bacterial removal efficiencies for storm water wetlands are 70% to 80%. The heavy vegetation slows water flow, allowing particulate sedimentation and infiltration to ground water. The standing water promotes physical, chemical, and biological processes. Well-designed and constructed wetlands are long lasting.

Vegetated swales are broad, shallow, terrestrial channels that often serve as substitutes for curb and gutter drainage systems. To operate effectively, swales need a shallow slope with thick vegetation growth. The underlying soil must provide adequate drainage to prevent accumulating standing waters.

There must be enough slope to promote water transport, but not so steep as to cause erosion and scouring; typical values are 2% to 4%. There are no reported measurements of microbial reductions in swales.

Among the more common BMPs, a wet detention pond is an excavated volume designed to capture and slowly release storm water runoff. The wet detention pond maintains a standing pool of water to promote physical, chemical, and biological processes to lower contaminant concentration in runoff. The local rainfall, ground water, and geology must provide a standing water pool. The standing pool typically provides sufficient residence time to promote solids settling and removal of particle-associated contaminants. The edges of the pond commonly have shallow ledges to promote plant growth for nutrient uptake, safety, and aesthetics. The pond design typically has an aspect ratio greater than about two to reduce short-circuiting. The controlled flow discharge reduces the hydrograph peak (Botts et al. 1996; Frederick et al. 1996). Pond sizing is based on the drained area and effective runoff coefficient. The runoff volume is typically modeled using simple hydrology techniques.

Installing buffer strips is a commonly prescribed BMP for protecting receiving waters from storm water runoff in agricultural areas, with *Cryptosporidium* often being the primary concern. Buffer strips, also called filter strips, are vegetated areas using single species or mixtures of grasses, legumes, or other forbs with stem spacing up to one inch installed parallel to the receiving water shore. Although experts debate the minimum required width, a commonly recommended minimum is about ten meters. The strip follows the contour, with variations less than 0.5%. The land slope immediately above the filter is typically 1% to 10% to ensure flow through and control maximum velocities. The adequacy of a buffer strip for protecting receiving waters is based on Natural Resources Conservation Service (NRCS) criteria. The NRCS National Handbook of Conservation Practices (NRCS 1997; NRCS 1998) contains the traditional filter strip design standards. Using these standards for pathogen control, NRCS expects a slight decrease in surface water pathogen contamination.

Moore et al. (1988) cite several studies that show the effectiveness of buffer strips in reducing nutrients and sediments in runoff. The mechanisms contributing to the effectiveness are reduction in volume from increased infiltration, decrease in velocity resulting in increased sedimentation of particulates with adsorbed pollutants, and increased pollutant adsorption to soil particles due to lower ionic concentrations. For a vegetated filter strip to remove sediment-bound organisms, it must provide an appropriate mechanism for removing sediment. Design procedures (Dillaha and Hayes 1991) identify several key considerations when selecting buffer strips. Filter strips are only effective under shallow sheet flow conditions. Sheet flow will occur if the filter strip can be installed approximately on the contour. Fields with extensive internal drainage concentrate surface runoff. Excessive sediment inflow to an effective filter strip will clog and shorten the useful life. Routine maintenance, e.g., mowing to encourage dense vegetation and weed control, inspection and repairs to fill gullies, removing flow-blocking sediment, reseeding, and other measures, prevents concentrated flow. Excluding livestock and vehicles reduces soil compaction and promotes infiltration. Walker et al. (1990) modeled the concentration of indicator bacteria in runoff resulting from a single storm event immediately after land application of waste. The model predicted that a 30-meter filter strip on a 3% slope could remove a maximum of 75% bacteria. The model did not show if increased length would result in further reductions.

Infiltration trenches capture and hold the runoff volume for infiltration. These devices are typically one to four-meter-deep excavations filled with aggregate and gravel installed in well-drained, low-sloped soils. Sufficient underdrainage is critical for proper operation. Sand filters can capture up to 90% of influent particulate matter (Botts et al. 1996). Functionally, infiltration trenches work as coarse-media sand filters discharging to ground water. While the ground water discharge replenishes ground water, there are often concerns about the remaining contaminants and areas with deep water tables. Maintenance is essential to prevent clogging as particulates accumulate in the filter media. More than half the installed infiltration trenches fail after five years from inadequate maintenance (Botts et al. 1996).

Source Water Protection and Watershed Management

EPA's Office of Water has defined SWP as a common-sense approach to guarding public health by protecting drinking water supplies. SWP measures prevent contamination and reduce the need for treatment of drinking water supplies. SWP includes managing potential contamination sources and developing contingency plans that identify alternate drinking water sources. A community may decide to develop an SWP program based on the results of a source water assessment, which includes the delineation of the area to be protected and an inventory of the potential contaminants within that area (USEPA 2000a). SWP from quality degradation by microbial contaminants (i.e., bacteria, protozoa, viruses, helminths, fungi) is any activity undertaken to minimize the frequency, magnitude, and duration of occurrence of pathogens or indicators (e.g., indicator microorganisms or turbidity) in source waters. SWP may also, by reducing the concentration of NOM, a DBP precursor, reduce the formation of DBP.

SWP strategies comprise the first stage in the multiple-barrier approach to protecting the quality of drinking water. Other major drinking water quality protection barriers include water quality monitoring and selective source withdrawal, water treatment processes for removal or inactivation of pathogens and control of DBP formation, water distribution practices for preventing intrusion or regrowth of pathogens, and point-of-use treatment where required.

SWP strategies are a specific subset of a larger watershed protection strategy applied when the protected receiving water is used as a water supply. Conceptually, watershed protection is heavily linked to pollution prevention, contaminant source identification, and risk management. Although watershed management does not have a universally accepted definition and connotes alternate approaches, each interpretation has an underpinning of holistic approaches to prevent or mitigate threats to the receiving water over a geographic region defined by a common hydrology.

Managing microbial contaminant risks in watersheds requires identification and quantification of organisms. Because of difficulties associated with assaying for specific pathogens, monitoring programs have tested for indicator organisms, including FCs and TCs, to identify possible fecal contamination in water. Monitoring regulations often specify indicators for determining water quality because the analytical methods are easier to complete, faster, and lower-cost than methods for specific organisms. Limitations of relying on indicators for determining the presence of pathogens include the occurrence of false positives. The indicators measure bacteria that live not only in human enteric tracts, but also in the enteric tracts of other animals (Toranzos and McFeters 1997).

Epidemiological studies in recreational waters (Dufour 1984) showed no correlation between measured FC densities and the occurrence of gastrointestinal illness in swimmers in fresh water, but a high correlation between gastrointestinal illness and *E. coli* and *Enterococcus* concentrations. Based on these results, EPA recommended that states adopt *E. coli* and *Enterococcus* as recreational water criteria in 1986, but some feel that these new indicators are inadequate (Calderon et al. 1991).

Methods to identify and quantify pathogens in watersheds require filtering large volumes of water and eluting the organisms from the filter. Detection and quantification are accomplished by culturing or molecular biology methods. Some organisms cannot be identified through culturing techniques, so molecular biology methods, based on nucleotides within nucleic acid sequences, are used. Low recovery efficiencies commonly encountered with filtration recovery make it difficult to estimate original concentrations with confidence. Methods for protozoa are cumbersome and do not indicate viability. Infectivity studies can be done to determine viability, but are expensive and slow. When an outbreak of a waterborne pathogen is suspected and the water is tested, the pathogen may not be detected because the contamination may have been temporary and been flushed out or died off (Moe 1997).

Modeling and Source Water Protection

Modeling can assist in identifying the vulnerability of a drinking water utility to threats from source water contamination. These models can be used in assessing the impact of upstream point-source discharges on downstream users as well as the potential for contamination from nonpoint sources (Clark et al. 1998). For example, the Water Supply and Water Resources Division (WSWRD) has developed two user-friendly modeling systems which include (1) a simplified model of the entire Ohio River, and (2) a detailed model of the Ohio River mainstream that may be used under emergency spill situations. Both models are built to interact with a Geographic Information System (GIS) for display and/or input generation, and it is anticipated that this approach will be extended to other source waters. The wide-scale model uses representative steady state flow regimes and represents movement by simple travel time relationship and transformations by dilution and decay mechanisms. Pollutants are routed through the RF1 reach file representation of the basin (Clark et al. 1998). The detailed mainstream model uses actual dynamic flow patterns as input to EPA's WASP4 water quality model (Ambrose et al. 1990). WASP4 is a dynamic compartment model that can be used to analyze a number of water quality problems. The Ohio River mainstream is represented in the model from a series of segments ranging in size from two to ten miles in length. The basic equation used in WASP4 governing decay of contaminants is as follows:

$$C_s = (M_g / Q_s) \exp(-k \bullet CT_s) \qquad (4\text{-}1)$$

where Q_s (L/s) is the flow in the segment, M_g is the mass of the pollutant (mg/s) that enters the segment, k is the decay coefficient with a typical value of 0.5/day, CT_s is the cumulative time of travel (days), "exp" denotes the exponential function, and C_s is the concentration in mg/L at the end of the reach. When the pollutant is stable and not reactive, the value for $k = 0$.

The detailed model includes a hydraulic model (the Corps of Engineers FLOWSED model), which has been combined with WASP4 to make spill modeling predictions. FLOWSED, which predicts daily flow quantities along the mainstream and portions of major tributaries near their confluence with the Ohio River, is applied daily by the Ohio River Division of the Corps of Engineers. Five-day forecast of stage and flow are generated for 400 mainstream and tributary segments, and the results were made accessible to the Ohio River Valley Water Sanitation Commission (ORSANCO) via telephone lines.

A relational database management system was used to organize the various sources of data used in the study. Individual data files included information on facilities, outfall, permit limits, monitoring data, and codes used in the other files. The NPDES permit number was used as the primary key in each of the files.

GIS modeling and Data Base Management System (DBMS) techniques were integrated into two tools for use by ORSANCO for analyzing spills in the Ohio River. The NETWORK component of ARC/INFO was used to provide a steady state contaminant routing capability. In addition, a C-based spatial decision support system was developed as a spill management system to serve as a quick response tool for analyzing and displaying the results of a pollutant spill into the Ohio River.

Research is underway to extend this modeling approach to microbiological discharges from CSOs. Research is also being conducted which is intended to extend this modeling concept to nonpoint source contamination.

Modeling Overland Migration of Pathogens

Another aspect of contamination modeling is the overland transport of pathogens. Although efforts to model overland transport of *Cryptosporidium* oocysts have been limited, such models are needed to

predict oocysts loads and estimate the effectiveness of management practices. This information may subsequently be used in reservoir models if accuracy requirements are met. Auer et al. (1998) identified the need for developing pathogen loading data in order to support pathogen fate and transport modeling within reservoirs. Several models exist that are capable of predicting soil loss, runoff, transport of contaminants from animal waste, and bacterial die-off. Whether these models, individually or combined, are capable of accurately predicting *Cryptosporidium* loads or reductions achieved during transport through buffer strips remains to be seen. Considerations in model selection include assumptions made by the model, the size of the watershed, availability of data, and the desired level of accuracy. The ability of a model to simulate oocyst transport is dependent on how well the model assumptions reflect the actual characteristics of oocysts and the landscape over which they travel. EPA has sponsored ongoing research to evaluate factors affecting overland migration of oocysts. A major goal of the research is to determine the degree to which oocysts tend to stick to different materials and then to evaluate their potential for runoff, either in attached or free-floating form. Key components of the project include jar tests to determine partitioning of oocysts among water, clay, or other soil, fecal matter, plant matter, etc.; flume tests to directly evaluate oocyst overland migration; evaluation and development of a modeling framework; and evaluation of protocols for measuring oocysts in high turbidity samples encountered in runoff samples.

Models used in predicting the transport of animal waste over land have typically utilized indicator bacteria. This approach is useful in assessing risk due to fecal contamination since indicator organisms are easily identified, while low levels of pathogens may not be discernable. Although FC is a common indicator organism for fecal wastes, its physical characteristics differ significantly from those of oocysts. This results in differences in die-off rates and soil retention. These differences result in inability of existing models to predict oocyst transport. Identifying and quantifying the mechanisms which affect die-off and retention of oocysts may facilitate the use of existing models for estimating oocyst concentrations.

Crane and Moore (1986) found that, of the several patterns followed during enteric bacteria die-off, the model for first-order die-off kinetics accurately described bacterial die-off under several conditions. However, the rate coefficient was highly variable due to differences in the effect of environmental factors on the assorted types of bacteria. The authors identified pH, temperature, solar radiation, moisture, application method, and application medium as critical factors in determining microbial survival. Information on the effects of these factors on oocyst survival is necessary in order to develop a die-off rate coefficient(s) for *Cryptosporidium*.

Reddy et al. (1981) combined an animal waste model with the Agricultural Runoff Management (ARM-II) Model to simulate the effects on the quality of runoff from land receiving animal waste. The microbiological submodel was developed by simulating FC die-off and retention in the soil. Moore et al. (1988) also developed a model, MWASTE, which follows indicator organisms from the animal waste through leaving the land as surface runoff utilizing bacterial indicator organisms. MWASTE is capable of including data on the slope and width of buffer strips. The model COLI (Walker et al. 1990) also examines the movement of indicator bacteria in runoff. Although these models contain a biological component, they cannot be used to predict oocyst transport. It is possible that newer models, with improved capability to predict hydrology and sediment transport, may be adaptable to predicting oocyst transport if mechanisms controlling overland flow were better understood. The New York City Department of Environmental Protection has conducted an evaluation including pathogen loading in its terrestrial models and determined that improvements in identification and quantification of oocyst sources was required (USEPA 1997).

Summary and Conclusions

Passage of the 1996 amendments to the SDWA has focused the attention of water utility managers and public health and regulatory officials on SWP and its role in protecting public water supplies. There is growing awareness that water treatment and/or disinfection may not always be enough to ensure the provision of potable and safe water to the consumer. The 1993 cryptosporidiosis outbreak in Milwaukee, WI, has raised the possibility that even water suppliers which meet all of the SWTR requirements of the SDWA are vulnerable (Okun et al. 1997; Fox and Lytle 1996).

Most utilities in the U.S. invest a great deal of time, energy, and capital in developing mechanisms for protecting against the impact of sudden changes in influent water quality. Some of these mechanisms include investment in excess capacity and development of emergency procedures (Miller 1989).

Concern over SWP is not limited to surface water supplies. Many ground water supplies have proven to be vulnerable as well, resulting in the various states implementing wellhead protection programs. Based on the 1996 amendments, the states will have to implement programs to decide if a system's source of supply is threatened as well as determine the means to prevent pollution. Communities will be allowed to ask for state assistance, and a certain percentage of the State Revolving Loan Fund has been earmarked to assist with SWP (Howell 1987).

The SDWA was passed in 1974 and amended in 1986 and 1996, but SWP under the SDWA actually began with the SDWA Amendments of 1986. The 1986 amendments included provisions for "Protection of Ground Water Sources of Water." Two programs were set up under this requirement: the "Sole Source Aquifer Demonstration Program," to establish demonstration programs to protect critical aquifer areas from degradation; and the "Wellhead Protection Program," which required states to develop programs for protecting areas around public water supply wells to prevent contamination from residential, industrial, and farming-use activities.

In the 1996 amendments to the SDWA, protection of source waters was given greater emphasis to strengthen protection against microbial contaminants, particularly *Cryptosporidium*, while reducing potential health risks due to disinfection by-products. This increased protection is embodied in the IESWTR (USEPA 1998). This rule applies to public water systems that use surface water or GWUDI and serve at least 10,000 people.

Two major threats to source water quality with respect to DBP control and microbial protection are natural organic matter and pathogens. As reflected in the previous discussion, the two pathogens which are currently of most concern are *Giardia* and *Cryptosporidium*.

Managing microbial risk requires identification and quantification of organisms. Because of difficulties associated with assaying for specific pathogens, monitoring programs have tested for indicator organisms, including FC and TCs, to identify possible fecal contamination in water. The potential sources of pathogens in source water are many and varied, including nonpoint runoff and discharges from treated and untreated sewage and combined sewer overflows. From a waterborne outbreak and public health viewpoint, both *Giardia* and *Cryptosporidium* are of primary concern. Monitoring regulations often specify indicators for determining water quality because the analytical methods are easier to complete, faster, and lower-cost than methods for specific organisms. Limitations of relying on indicators for determining the presence of pathogens include the occurrence of false positives. The indicators measure bacteria that live not only in human enteric tracts, but also in the enteric tracts of other animals (Toranzos and McFeters 1997).

Microbial pathogens are found in treated sanitary sewage and wet weather flows, i.e., SSOs, CSOs, and storm water runoff. Many factors effect the types of organisms found and the concentrations at which they are detected. These include watershed contributions, treatment plant efficiency, and length of antecedent dry weather period. These treatment technologies can be both sources of contamination as well as protective of source water quality. In addition to the installation of sewage treatment and combined overflow systems, there are passive pollution prevention and mitigation techniques called BMPs. The techniques vary dramatically in application, ranging from social practices to engineering applications.

SWP strategies are a specific subset of a larger watershed protection strategy applied when the protected receiving water is used as a water supply. Conceptually, watershed protection is heavily linked to pollution prevention, contaminant source identification, and risk management.

Modeling can assist in identifying the vulnerability of a drinking water utility to threats from source water contamination. These models can be used in assessing the impact of upstream point-source discharges on downstream users as well as the potential for contamination from nonpoint sources (Clark et al. 1998). Another aspect of contamination modeling is the overland transport of pathogens. Although efforts to model overland transport of *Cryptosporidium* oocysts have been limited, such models are needed to predict oocyst loads and estimate the effectiveness of management practices.

Although SWP is currently more of a collection of practices than a well-defined art or science, it is anticipated that it will become an integral part of water treatment practice in the future. As interest grows in the concept of watershed management, it is likely that interest will grow in understanding the factors that effect the quality of source water for drinking water utilities as well.

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